A Critical Analysis of Methodologies Evaluating Biodiversity in Offset Banking

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Abstract: The assessment of ecological impacts represents a key element of biodiversity offsetting success. After briefly introducing the main controversies arisen on biodiversity monetization and defining biodiversity banking schemes, discussions will be focused on evaluation methods mobilized in the context of biodiversity offset banking. Although there is currently no fixed framework assessing environment due resolutely to the specific nature of biodiversity, evaluation methods are analyzed through a review of the academic and empirical literature. This step allows to select few methods justifying these choices and presenting their pros and cons by keeping the aim to contribute to the debates. Thus, ecological assessments (service-to-service and resource-to-resource) and economic valuations (value-to-value and value-to-cost) carried out in biodiversity banking schemes will be distinguished. Finally, this paper will emphasize the inherent differences of the two evaluating forms and their specificities on the one hand, and highlight their opportunities and risks from methodological perspective on the other hand.

Key words: Biodiversity offset, offset banking, biodiversity evaluation, ecological assessment, economic valuation.

1. Introduction

Biodiversity offsets are conservation activities that are designed to create biodiversity gains to compensate for losses according to a mitigation hierarchy process. The goal of biodiversity offsets is to achieve NNL (no net loss), or preferably a net gain of biodiversity on the ground with respect to species composition, habitat characteristics and ecosystem services. Three major periods can be identified in the development of biodiversity offsets [1]. The first period (1950s to 1970s) involved the idea of compensation that was promoted by wildlife specialists and supported by lawyers. Their wish to provide local and ecological equivalence led to the creation of in-kind compensation or in-kind offsets. This was intended to restore ecosystem functions from an ecological point of view by means of restoration, rehabilitation, creation and/or preservation to offset environmental damage [2]. The second period (1970s to 1990s) involved the emergence of compensatory measures under the influence of economists. In some countries, financial transfers or financial compensation represent an alternative to in-kind compensation: a developer pays an amount to environmental entities, but does not need to justify the measures resulting from this payment. Depending on the legislation of the countries concerned, monetary payments may be accepted in exceptional cases as a last resort, or they may be tolerated after in-kind offsets, or they may be considered as the sole method of compensation. These transfers have several limitations such as the risk of relieving the project developer from responsibility, of producing...
A Critical Analysis of Methodologies Evaluating Biodiversity in Offset Banking

non-equivalent offsets, of underestimating the sums required to implement and manage the offset, of not allocating sufficient funding or a lack of additionality [2]. In the current phase (since the 1990s), confronted with the difficulties encountered with the direct implementation of offsets by the project developer, compensation banks or offset bank systems have emerged in some countries (US, Australia, Germany and France). In 1987, the American National Wetland Policy Forum used “mitigation banks” for the first time, and expressly stated that this instrument is consistent with the NNL objective. These mitigation banks create a supply of “mitigation units” to make it easier for project developers to implement offset measures. They were at first devised by public agencies and private firms to meet their own needs for wetland mitigation measures. Since then, landowners and companies have gradually become involved in the process. Biodiversity banking involves transposing mitigation banking to the conservation of endangered species, habitats, environment and/or general biodiversity. It is intended to create an effective or potential supply of offsets for certain specific environments, habitats or species in advance, in order to meet the compensation needs of future projects. By generating a supply of “biodiversity units”, biodiversity banking often reduces the time-gap between the environmental damage and compensatory measures in order to achieve the NNL requirement.

This paper aims at analysing the third development phase of biodiversity offsetting, e.g., banking schemes, to identify and discuss methodologies evaluating biodiversity or ecological impacts in these mechanisms. This paper contributes to the debate by summing up and enlarging the classifications made by the Environmental Law Institute [3, 4], Robertson and Hayden [5], and Hough and Robertson [6]. Five key types of biodiversity banks are differentiated here and defined the following mapping approach [7]:

- Single-user banks are banks in which the promoter is also the client. Agencies (e.g., the State Agencies of the Federal Department of Transportation in the US) and industrialists can establish their own bank to enable them to offset several of their own projects. This banking scheme is developed in the US4, in Madagascar (Rio Tinto), etc.;
- A private entrepreneur manages private commercial banks whose biodiversity credits are available for sale on the market. They are independent of the project developers. The customers of such banks may be public or private entities. These banks now make up almost 70% of mitigation banks in the US (on wetlands and on habitats/species) [2]. They are also developed in the Netherlands;
- Hybrid commercial banks are set up by agreement between private and public entities. The promoter of the biodiversity credits is private and the government acts as both the regulator and the broker. Potential customers may be project developers seeking to offset their projects’ impacts on biodiversity, conservation groups or government agencies. These banks are essentially implemented in Australia;
- Public commercial banks are administered by public entities to offset impacts caused by either public or private development projects. This banking scheme is implemented in France and partly in the UK (United Kingdom) (even if it may be applied with some characteristics from hybrid commercial banks);
- Public non-commercial banks generate units solely for use by various public agencies. A single or a consortium of public entities, such as Federal, State, and/or local government agencies, sponsors the bank. This kind of biodiversity banks is implemented in Germany and represents a reference for lot of countries that would like to include offset since the planning step.

4It is essential to note that the US is not only developing non-commercial single-user banks (private or public). Indeed, private commercial and public commercial banks could also be identified in different states.

5All BB (biodiversity banks) schemes have made offsets before the implementation of project developers. On the contrary, other compensatory measures do not expressly require making offsets before the project is executed.
Thus, this approach illustrates the distinction between single-user and multiple-user banks, between private and public banks, and between commercial and non-commercial banks. Commercial banks make it possible to develop the trade in environmental assets (biodiversity credits), whereas non-commercial banks reject this “profitable trade” approach, and generate “biodiversity units” corresponding to whatever form of “biodiversity” is targeted by the bank (habitat, species, functions and/or services).

This paper targets to identify and discuss methodologies valuating biodiversity in the context of this mapping approach of BB (biodiversity banks). To enforce compensatory measures, offset banking measures have to evaluate biodiversity and be able to use the more adapted tools realizing environmental equivalence. Currently, there is no established method available to evaluate biodiversity or information on their use. Even if these tools are exposed in national recommendations that commonly established, in fact, a hierarchy on approaches to use for reaching the NNL state [8, 9], methodologies evaluating biodiversity face some practical difficulties to obtain equivalence. This notion of equivalence is closely linked to evaluation because biodiversity banking schemes are implemented to reach the ecological equivalence using valuation methods intended to compare the injured biodiversity to the compensated biodiversity. Thus, this paper adopts a progressive approach successively highlighting the valuation methodologies mobilized by biodiversity banks and their advantages and limitations. This approach is analyzing in what way biodiversity valuations are used by biodiversity banking schemes, which methods are mobilized in what context and for what purpose.

To assess biodiversity, plenty of methodologies have been implemented because of the impossibility to find a unique and accepted benchmark comparing injured and compensated biodiversity. This fact contrasts with the carbon compensation and its ton of carbon dioxide equivalent. Thus, to develop a critical and cross-analysis of methodologies evaluating biodiversity values, this study is based on academic studies [10-14] and institutional literature [15, 16]. This work leads to retain two principal forms of assessment: the ecological and the economic one.

Section 2 considers some theoretical and methodological considerations on biodiversity assessments in offset banking; Section 2.1 analyzes two principal “ecological assessment” methodologies used to estimate biodiversity in single-user banks and non-commercial banks; Section 2.2 deals with the “economic assessment” methodologies that are mostly used by commercial banks (private, hybrid and public) to value biodiversity and attribute prices to biodiversity units; Section 3 presents results and discussion on the different methodologies used to assess ecological impacts in biodiversity banking; Section 3.1 develops results and discussion on ecological assessments used by non-commercial banks; Section 3.2 develops results and discussion on economic assessments used by private, hybrid and public commercial banks. This comparative and critical analysis will contribute to clarify biodiversity banking and associated methodologies evaluating biodiversity; Finally, Section 4 gives some conclusion and the direction of future works.

2. Theoretical and Methodological Considerations

2.1 Ecological Assessments: Some Theoretical and Methodological Considerations

EAs (ecological assessments) were developed in the US since the 1950s by biologists and ecologists to improve awareness and knowledge on biodiversity. With the development of economic tools for environmental conservation, the debates of environmental evaluation have prompted some ecologists and biologists to rethink what they do and
A Critical Analysis of Methodologies Evaluating Biodiversity in Offset Banking

experiment with new objectives. Though, two major methodologies evaluating biodiversity were implemented until the 1990s and highlighted by the NOAA (National Oceanic and Atmospheric Administration) since 1995 [15]: S-S (service-to-service) and R-R (resource-to-resource) methods. Both of them use proxies to evaluate environmental losses and gains. Non-commercial banks such as private or public single-user banks (e.g., American Ministry of Transportation or private international firms) and public umbrella banks (e.g., in Germany) use these EAs to create biodiversity units used for their personal compensation needs. Generally, the first banks developed in industrialized countries implemented single-user banks to be pro-active in terms of legislation, to improve their image (societal responsibility) or to fulfill recommendations of international funding agencies (e.g., World Bank).

In order to estimate the equivalencies between losses and gains of ecosystem services, the HEA (habitat equivalency analysis) has been selected by non-commercial banks from among the equivalence methods that can be used to evaluate compensatory measures [17, 18]. The HEA has been used in NRDA (natural resource damage assessments) under the Comprehensive Environmental Response, Compensation and Liability Act and the Oil Pollution Act [19-21]. This methodology is based on a non-monetary approach, calculating damage and compensation in biophysical units; it could be placed in the S-S equivalence method.

First of all, in S-S method, the proxy used to illustrate injured biodiversity may be a biological indicator representing the injured ecosystem (vegetative cover, presence of endangered species...) or species in the case of significant links with other species (furniture of ecosystem services) [22]. It could also be a composite indicator made up of several resources and/or services. The HEA developed by the NOAA in 1995 illustrates this methodology [15]. Gains and losses are quantified as "hectare-service-years", then discounted and expressed in DSAYs (discounted service acre-years). A widely accepted 3% discount rate is used, so as to make past and future gains and losses comparable. The HEA was summarized in equation [20-24] as follows:

\[ V_I A_I (1 + r)^{-t_I} = V_R A_R R (1 + r)^{-t_R} \]  

(1)

where, \( V_I \) is the value of the ecosystem or function impacted, \( V_R \) is the value of the ecosystem or function compensated, \( I \) is the intensity of impact, \( R \) is the intensity of compensation, \( -t_I \) is the time-scale of the impact, \( -t_R \) is the time-scale of the compensation, \( r \) is the discount rate, \( A_I \) is the number of acres impacted (damaged area) and \( A_R \) is the number of acres compensated (compensatory area). All these values are built by inventories, taxonomies, ratios implementation and summaries made by wildlife specialists.

Thus, the HEA is aiming at calculating the size of the compensatory measure \( A_R \) while illustrating the most spread form of the S-S methodology. The HEA intended to quantify the losses due to the damage in the injured area by showing the shaded damage illustrated by their NPV (net present value) in the left side of Eq. (1). Having the surface of the injured area, biologists and ecologists will then try to determine the surface area needed to compensate for damage. By quantifying the gains obtained from the banking site, researchers will be able to find the right side (NPV) of Eq. (1). Once obtaining these both data, they are able to determine the size of the biodiversity-banking project by multiplying the evaluated compensatory area to reach the needed biodiversity level.

The HEA provides a way to evaluate species in a manner that correlates ecological function of all
habitats before and after compensatory measures, and quantifies the improved ecological function achieved through offsets. Therefore, biodiversity banking might use HEA to highlight the improvement of biodiversity units while using the values calculated before and after measure implementation. Even if this method was implemented in the 1990s, service-to-service is still used particularly in the US and European countries. This case-by-case assessing method allows making precise ecological assessment in application to different habitat types. Public non-commercial banks often use this method to underline the quality of their biodiversity assessments [7]. It has already been successfully used on habitat-scale, in-kind restoration of seagrass [25] and salt marsh [26] habitats. To extend the application and speed up S-S method, Kohler and Dodge [27] developed software (Visual_HEA) to calculate compensatory restoration following natural resource damage. This operational simplification allows using ecological assessments in a larger scale.

Concerning the R-R method, the most extended valuation model is the REA (resource equivalency analysis). This method is relatively expensive and relies primarily on biological information collected in the course of determining natural resource damage. It is consistent with approaches recommended by northern competent authorities in national guidelines but takes lots of time. Non-commercial banks in the US (public banks) or in Germany (single-user of umbrella banks) use this method as well as S-S method. REA involves determining the amount of “natural resource services” [28] that the affected resources would have provided if not been injured, and it equates the quantity of lost services with those created by proposed compensatory restoration projects that would provide similar resources (species, habitats...). The unit of measure may be acre-years, stream feet-years, or some other metric. As generally made in ecological assessments, future years are discounted at 3% per year, consistent with NOAA recommendations for natural resource damage assessments [16]. These calculations may be done in a variety of ways, but the most common are to estimate the damage and the restoration benefits in terms of area years of habitat or animal years.

2.2 Theoretical and Methodological Considerations on Economic Assessments

Two major economic methodologies evaluating biodiversity have been identified: V-V (value-to-value) and V-C (value-to-cost) methods. Private, hybrid and public commercial banks should attribute economic value to each hectare to be able to sell biodiversity credits to developers.

Starting with the notion of TEV (total economic value), theoretical and methodological characteristics of V-V approach aiming at equalizing values of biodiversity loss and biodiversity gain were highlighted [29]. This approach is based on the monetary evaluation of biodiversity according to Costanza et al. [30] evaluation process. The idea is to attribute economic value to biodiversity, e.g., use value and non-use value that is underlying the TEV (Fig. 1) by direct valuation method (also named contingent valuation method)7. The National Oceanic and NOAA proposed it in 1993 [31]. It aims at collecting declared preferences (individual WTP) gathered from surveys. Many ecosystem services are not traded in markets so people affected by environmental damage may never state what they are willing to pay for them. Simply asking them what they

7 Other methods based on revealed preferences attribute economic value to the damaged/restored site through an analysis of people WTP (willingness to pay) [32, 33], e.g., when people purchase something (a home near a wetland) or spend time and money to get somewhere (a fishing spot or bird watching dependent on a nearby wetland). These methods only valuate direct use value. The last accepted approach for estimating economic value within V-V method is the “derived willing to pay” method that traces and measures the functions provided by an ecosystem (e.g., retaining floodwater, reducing wave energy, maintaining water quality) and estimates how much people affected by the damage would be “willing to pay” to avoid the adverse effects of losing them. The dollar value if flood and siltation damage avoided because of a wetland is an example of derived WTP for ecosystem services [34].
would be willing to pay to avoid environmental damage or to benefit from environmental improvement can sometimes yield useful results.

Here, V-V method was linked to private and hybrid commercial banks [7] even if this method is gradually disappearing. This method was employed by a little number of the first public commercial wetland banks in the US appeared few years after the first public single-user banks. Nowadays, V-V method is only sporadically used in the US and in Australia. The Europe refuses to value biodiversity with this approach for its controversies. This method is used when the injured and restored resources and services are not of the same type, quality, and value in order to calculate the value of gains from the proposed restoration actions and the value of the interim losses. This method largely based on direct (or contingent) valuation is more and more left out because of its long-lasting specificity (precise inventories and long commodification process) and costliness (to pay experts and monitoring). Although V-V has been widely discussed for the past two decades (i.e., controversies on Costanza et al. [30] results), there is considerable controversy over whether it adequately measures the WTP of bankers for environmental quality and NNL achievement. Surveys of WTP are expensive and controversial and usually yield results that are reliable only when questions are asked about specific wetland services provided in specific contexts. In practice, a notable dominance of direct and indirect use values added with non-use values might be observed. The lack of option value assessment in biodiversity banks is due to scientific difficulties to inventory, list and forecast all the biodiversity (genes, functions/services, etc.).

To reduce critics against V-V approach, more and
more commercial biodiversity banks (public, hybrid, and private) have been using value-to-cost method to assess biodiversity. V-C is carried out to valuate biodiversity from economic point of view by adding all costs included in the implementation and the management of compensatory measures. It aims at balancing the economic value of service loss and the total cost of compensatory measures [14]. In the case of hybrid commercial banking, an economic value is associated to each biodiversity unit—usually corresponding to one hectare of selected biodiversity (species, habitats, ecosystem functions...)—to create biodiversity credits. This monetary value includes all costs such as creation, rehabilitation, preservation and/or restoration measures to compensate for injured biodiversity (introduced species or replanted varieties, habitat creation, idiomatic species’ introduction...), the cost of land acquisition or rental for establishing the bank, and operating costs of the bank (wages, maintaining and enriching offset measures...). In a number of situations, the bank also bills its offsetting service to developers.

A final category of economic assessment method is known as BT (benefits transfers) or values transfers. This methodology is based on extrapolations and refers to the use of estimates obtained (by whatever method) in one context to estimate values in a different context. This approach is more and more used by commercial banks with valuation accumulation. For example, an estimate of the benefit obtained by tourists viewing wildlife in natural area might be used to estimate the benefit obtained from viewing wildlife in compensated area. Alternatively, the relationship used to estimate the benefits in one case might be applied in another, by using adjusted data from this case in conjunction with some data from the site of interest (“benefit function transfer”). For example, a relationship that estimates tourist benefits in one landscape, based in part on their attributes such as income or national origin, could be used in another landscape, but with data on income and national origin of that landscape’s visitors. Another use of value transfers by hybrid commercial banks aims at reducing valuation expenditures on comparable habitats, genes or ecosystem functions. Because of the unwieldy evaluating structure and methodologies [36], transfer value technics were progressively implemented using known values to apply them on comparable biodiversity, where assessments are difficult. In the last years, lot of literature aims at clarifying the conditions needed to valid this approach.

3. Illustrations and Critical Analysis

3.1 Illustrations and Critical Analysis of Ecological Assessments in Biodiversity Banking

S-S method is illustrated by the example of the single-hull tanker Athos I. This tanker, registered under the flag of Cyprus, was reported to be leaking oil into the Delaware River to its terminal at the CITGO asphalt refinery in Paulsboro, New Jersey. Non-commercial banks for its pedagogic development largely retake this example. Even if these biodiversity banks do not publish their entire studies for confidential reasons, they explain to use traditional ecological assessments.

Thus, in the Athos I example, the anchor punctured the vessel’s bottom, resulting in the discharge of more than 263,000 gallons of crude oil into the Delaware River and nearby tributaries (in wetland areas). Under the federal OPA (oil pollution act), two federal government agencies—the NOAA and USFWS (US Fish and Wildlife Service)—and the three affected states—New Jersey, Pennsylvania, and Delaware—are responsible for restoring natural resources injured by the Athos spill. This measure could be assimilated to a single-user bank using in-kind compensation. The two federal agencies and the three affected states, acting as Trustees on the public’s behalf, have conducted a NRDA to determine the nature and extent of natural resource losses resulting from this incident and the restoration actions needed to restore these losses. The
NRDA was conducted using the OPA NRDA regulations. Both the damaged and the compensation sites should be evaluated to be able to implement compensatory measures to reach the NNL. Scaling calculations include both direct and indirect damage (i.e., direct mortality from the spill as well as indirect mortality due to lost productivity). Damage is scaled by guild based on approximate weight and diet of the birds and other indicators used in this study concerning metric of ecosystem services are not presented. However, the additional losses of DSAYs could be found by adding all the DSAYs calculated for each impacted element (discounted at 3% per year). An abstract of the damage assessment results, as described in the preceding theoretical part, is provided in Table 1.

Table 1 should be completed by another abstract of the compensation restoration acreage by habitat type developed in the same report (Table 2).

Thus, all damage to habitats could be summed up in the category “shores”, that is to tell seawalls (27 DSAYs), sand/mud substrates (32 DSAYs), marsh (54 DSAYs), lower intertidal zone (46 DSAYs), tidal flats (929 DSAYs) and coarse substrates (114 DSAYs) was offset by the restoration of 34.2 acres of marsh brackish coastal, and by the restoration of 0.9 acres of meadows and wetlands freshwater near the affected area (Table 2).

In another example of HEA application used in S-S method, Peterson and Associates [37] estimated the area of habitat necessary to replace un-vegetated, estuarine bay bottom and the associated water column sacrificed as part of an expansion of the Craney Island Dredged Material Placement Area on the Elizabeth

<table>
<thead>
<tr>
<th>Resource damage category</th>
<th>Resource</th>
<th>Damage estimate</th>
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<tbody>
<tr>
<td></td>
<td></td>
<td>Acres or trips</td>
</tr>
<tr>
<td>Shoreline</td>
<td>Seawalls</td>
<td>59.38</td>
</tr>
<tr>
<td></td>
<td>Sand/mud substrate</td>
<td>1,415.83</td>
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<tr>
<td></td>
<td>Coarse substrate</td>
<td>137.23</td>
</tr>
<tr>
<td></td>
<td>Marsh</td>
<td>116.47</td>
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<tr>
<td></td>
<td>Tributaries</td>
<td>1,899.23</td>
</tr>
<tr>
<td>Aquatic</td>
<td>Subtidal benthic habitat</td>
<td>412</td>
</tr>
<tr>
<td>Bird and wildlife</td>
<td>Dabbling ducks, diving ducks, diving birds, gulls, shorebirds, wading birds, swans/geese, kingfishers</td>
<td>20,027.5 kg of bird lost</td>
</tr>
</tbody>
</table>

Source: NOAA et al. 2009 [21].

<table>
<thead>
<tr>
<th>Habitat classification</th>
<th>Acres</th>
<th>Adjusted DSAYs(^a)</th>
<th>Marsh DSAYs(^b)</th>
<th>Marsh restoration acres(^c)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marsh</td>
<td>117</td>
<td>54</td>
<td>54.0</td>
<td>4.0</td>
</tr>
<tr>
<td>Sand/mud substrate</td>
<td>36</td>
<td>32</td>
<td>12.6</td>
<td>0.9</td>
</tr>
<tr>
<td>Lower intertidal zone</td>
<td>83</td>
<td>46</td>
<td>18.4</td>
<td>1.4</td>
</tr>
<tr>
<td>Tidal flats</td>
<td>1,298</td>
<td>929</td>
<td>371.5</td>
<td>27.7</td>
</tr>
<tr>
<td>Coarse substrates</td>
<td>137</td>
<td>114</td>
<td>11.4</td>
<td>0.9</td>
</tr>
<tr>
<td>Seawalls</td>
<td>59</td>
<td>27</td>
<td>2.7</td>
<td>0.2</td>
</tr>
</tbody>
</table>

\(^a\)Adjusted DSAYs take into account the baseline damage estimate for the Delaware River (a 10% ecological service loss);

\(^b\)Marsh DSAYs are calculated by dividing DSAYs by habitat equivalency factor (1 for marsh; 2.5 for sand/mud substrates, intertidal and tidal flats; and 10 for seawalls and coarse substrates);

\(^c\)Marsh restoration acres are calculated by dividing marsh DSAYs by the weighted average per-acre credit for restored marsh (13.4 DSAYs/acre).

Source: NOAA et al. 2009 [21].
River, Virginia. Peterson and Associates [37] based their estimates on lost secondary production of herbivores (in fauna and zooplankton) and made separate estimates for potential replacement of the bay bottom and water column habitat by either oyster reef or salt marsh habitat. In order to offset total losses for the 234 ha (580 acres) site, they estimated that it would require between 2.0 ha and 7.4 ha (5.0 acres to 18.2 acres) of oyster reef habitat or 27.0 ha to 98.2 ha (66.9 acres to 243.2 acres) of salt marsh habitat. In this case, the authors recommended that a mix of oyster reef and salt marsh habitat be constructed, since the combination was likely to provide synergistic ecological benefits.

To illustrate the R-R method and its main extended valuation model (REA), the following example is taken. In US, when non-commercial biodiversity banks are created to offset damage concerns primarily individual animals rather than a complete habitat, the REA may focus on lost animal-years. For example, supposing an oil spill causes negligible damage to a body of water, but results in the death of 100 ducks. Using information about the life history of the ducks (e.g., annual survival rate, average life expectancy, average fledging rate, etc.), wildlife specialists can estimate the “lost duck-years” due to the spill. On the gain side, they examine restoration projects designed to create duck nesting habitat, and scale the size of the project to create as many duck-years as were lost in the incident.

In discussion, S-S and R-R methodologies are approximately comparable in term of assessment of ecological impacts even if R-R method contains more details and is addressed to wildlife specialists. From institutional perspective, there is no clear difference between these methodologies but differences in terms of practices and society expectations. Indeed, according to the development of biodiversity offsets, biologists and other wildlife specialists initiated the environmental evaluation. The result of this historical evolution is the rigorousness developed in R-R method in first times, and the interest for environmental services/functions of S-S approach (and then the ecosystem services [39]). That is why this paper indifferently uses EA to speak about S-S and R-R evaluations. In terms of frequency of use, S-S method is more often used than R-R assessment for time and practical reasons. Indeed, being more details than S-S method, R-R valuation takes more time and money to lead to more uncertain information than the other method. This uncertainty is largely linked to the lack of scientific knowledge on species, habitats, ecological connectivity, etc.. To reduce the gap between environmental needs and recommendations, some approximations are made by using proxies and ratios. In addition, it is important to say that EA are the basis of economic valuations that need biological inventories and ecological knowledge. S-S approach has a number of advantages for calculating restoration ratios and therefore environmental benefits. It also has a number of limitations. Dunford et al. [20] summarized them pointing out that underlying assumptions are frequently impossible to achieve (injured and restored habitats will eventually produce the same quantity and quality of services), and proportion of habitat services to habitat values is supposed constant over time. To improve these weaknesses, Thur [40] proposed a modification in HEA and REA methods to make it more amenable to permanent losses caused by approved development projects.

Then, ecological assessments have some advantages. Firstly, these methodologies ink the study in a strong sustainability approach because equivalences in terms of physical units are required [41]. Secondly, because of its general way of application, these methods can be used on a variety of ecosystems. Thus, ecological assessments might be a good approximation for restoration measures offsetting environmental damage [14]. Thirdly, these approaches aim at restoring
biodiversity with reasonable costs. So, ecological assessments are contributing to choose the best environmental restoration for a given amount thanks to strong environmental analysis and assessments. For this last reason, some of national guidelines recommend to use ecological assessments completed with economic ones to reach as fast as possible the NNL.

Despite these advantages, ecological assessments are still barely used on both the affected and compensated sites to see if the species, habitats and/or ecosystem functions have been recreated enough to achieve NNL. For some scholars as Robertson [42], the choice of compensation ratios should not be based on previous studies, but be reassessed on a case-by-case basis. In addition, ecological methodologies evaluating biodiversity suffer from a global lack of knowledge in biodiversity comprehension. This led some dissimilarities in results generated by differences in metric choices [43] between wildlife specialists. Beyond the limits linked to the restrictive character of hypothesis (accuracy of the results, stable evolution of services...), ecological assessments do not take sufficiently into account ecological connectivity or dynamic evolution of ecosystems [44]. The last key pitfall of ecological assessments lies in the difficulties to evaluate all ecosystem services described by the MEA (millennium ecosystem assessment) [39]. Indeed, an underestimation of environmental impacts seems to be systematically realized for theoretical and practical reasons [45], such as simplifications, necessary time, scaling, and metric uses. Even if an assortment of methods for rapid assessment of ecosystem functions has been developed and tested [46], in the absence of an accepted method, losses and gains are primarily accounted for in terms of area of specific ecosystem and associated vegetation [42]. Currently, practices are not enough spread to be able to create systems such as the RAM (rapid assessment method) because knowledge is not sufficient to provide standardized and replicable estimates [5].

Even if these two ecological assessments get some dissimilarities, they are both used by non-commercial banks [7] and may be completed by monetary valuations for political and/or communication reasons. Thus, to complete ecological assessments or to create biodiversity credits and add flexibility within biodiversity banking process, environmental agencies, think tanks, NGOs (Non-Governmental Organizations) and bankers progressively implement and improve economic methodologies evaluating biodiversity.

3.2 Illustrations and Critical Analysis of Economic Assessments in Biodiversity Banking

As an illustration of V-C method, this paper develops the French example and the NOAA V-C methodology. Table 3 shows the amount of restoration project costs, such as what was made in south of France in the CDC (Caisse des Dépôts et Consignations) Biodiversity [7]. If an average is roughly made, one acre of biodiversity is sold for USD77,250. The price of biodiversity depends on urban pressure, biological scarcity, and so on, that is why prices are very different. Indeed, developers should select the best biodiversity credit to offset damage with similar biodiversity while reducing the cost. The NOAA also highlighted this method [16]. This example and the Table 3 allow to illustrate how costs are allocated between the several elements of commercial biodiversity bank implementation and management.

Table 3 represents costs for 25.4 acres of wetlands, 35 acres of wet meadows and 100 acres of grassland habitat restoration. Grassland restoration costs are

<table>
<thead>
<tr>
<th>Cost element</th>
<th>Total cost (USD)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Planning and design</td>
<td>315,586</td>
</tr>
<tr>
<td>Construction</td>
<td>11,213,713</td>
</tr>
<tr>
<td>Monitoring</td>
<td>628,640</td>
</tr>
<tr>
<td>Operations and maintenance</td>
<td>233,006</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>12,390,945</strong></td>
</tr>
</tbody>
</table>

Source: NOAA 1997 [16].
Total*: total project costs do not include contingencies of 25%.
included in the unit costs for wetland and wet meadows restoration. Grassland restoration is an essential project component and would take place even in absence of damage that can be scaled to it, as it serves as a means for on-site, upland disposal of excavated sediments. Contouring and revegetation of such excavated sediments is standard practice.

To quote another example, using V-C method, ecosystem marketplace has been presenting the American national range in credit prices. In 2008, this American range in credit prices was USD3,000 to USD653,000, with the average price at USD74,535. When tidal or vernal pool credit prices are included, the average is USD112,449. The highest price was recorded in Virginia, and the lowest in Arkansas. For stream credits, the national range in 2008 was USD15 to USD700, with an average price of USD260. The unpredictability in the market-value credits reflects differences in the availability and price of land appropriate for bank development and the cost to create an acre of wetland compensatory units within different regions (ecosystem marketplace): USD24,000 to USD46,000 per acre of non-riparian wetland in north Carolina, USD36,000 to USD63,000 per acre of riparian wetland in north Carolina, USD156,000 per acre of coastal wetland in north Carolina, USD55,000 to USD65,000 per acre of nontidal wetland in southeast Virginia, USD125,000 to USD150,000 per acre of nontidal wetland in northern Virginia, USD400,000 to USD653,000 per acre of tidal wetland in Virginia and USD84,500 per acre of wetland in Oregon. As noted above, credits in different regions of the US use different metrics (acres, fractions of an acre, or ecosystem function) to calculate credits. Therefore, the credit price in one region cannot be directly compared to a credit price in another region. Nevertheless, presenting these results gives an idea of credit prices of various ecosystems and habitats. Other commercial banks from industrialized countries such as France, UK, Canada, etc., use V-C valuation. However, the calculation details are rarely published for confidential and commercial reasons.

To illustrate the last economic assessment method, e.g., BT or values transfers, this paper provides the following example. An environmental group presents testimony in Oregon based on a widely disputed study in Louisiana that generated a wetland economic value of USD28,000 per acre. After disputing the validity of the estimating method and of using estimates from Louisiana in Oregon, the opposing side agrees to accept the number as fact, and points out that the county already requires USD40,000 per acre in compensation for wetland impacts as part of its wetland mitigation banking scheme. Later in the year a group of wetland developers who are also paying USD40,000 per acre as wetland impact fees sue the state to reduce the fee and, using evidence presented by the environmental group, get the fee lowered to USD28,000. In the use of value transfers, having a large scale of inventories and scientific knowledge is important [47]. Nevertheless, benefits transfers method has been the subject of considerable controversy in the economics literature, as it has often been used inappropriately. According to the MEA [48], a consensus seems to be emerging that benefits transfers can provide valid and reliable estimates under certain conditions [49]. These conditions include the requirement that the commodity or service being valued be very similar at the site where the estimates were made and the site where they are applied and that the populations affected have similar characteristics. Of course, the original estimates being transferred must themselves be reliable in order for any attempt at transfer to be meaningful. But as the conditions at the two sites are unlikely to be perfectly identical, some transfer error is to be expected. This feature, however, does not speak as such against the application of benefits transfer in real-world decision-making. This is because estimates based on benefits transfer can be generated with considerably less time and resources than primary studies. In a world of scarce resources
and typically very costly primary studies, decision makers may be willing to trade quick and cheap numbers against a certain loss in accuracy, provided that minimum quality standards are met.

In discussion, economic assessments have several advantages and limits on the ground. V-V and V-C are economic methods contributing to turn biodiversity units (resulting from ecological valuations or taxonomies) into biodiversity credits using economic valuations. Even if V-V approach is not hugely expended, V-C is increasingly used by commercial banks. In light of its applicability and relative rapidity, V-C allows commercial banks to sell quickly biodiversity credits and reinvest incomes in new compensatory measures pursuing the improvement of biodiversity units. Dealing with biodiversity banking effectiveness, commercial banks seem to be able to manage biodiversity units, although ecosystems, habitats and species might be underestimated with this method [50].

This commercial status creates a market-based trade of biodiversity units that can contribute to find economic means to mitigate biodiversity.

Moreover, economic methodologies (V-V, V-C and BT) valuing biodiversity offer the advantage of making one monetary dimension summarizing all dimensions of environmental impacts [51]. This common yardstick makes it possible for politicians and decision-makers to discuss on compensatory measures and integrate recommendations as soon as possible in processes. Despite these opportunities, some pitfalls may be highlighted. Indeed, economic valuation in V-V approach would therefore require accepting questionable hypothesis [52] such as the consumer capacity to estimate prices, its willingness to state real prices, the context influence on responses, the existence of substitutable goods or service. In addition, economic assessments are currently still suffering from both a little operational scope and a double-counting risk [42, 53]. The little operational scope of economic valuations is largely linked to its dependence on previous studies. The double-counting risk is associated with the mobilization of several assessment methodologies. Indeed, if genes or species are valued by a first inventory, the evaluation of ecosystem services may assess several time the same element for its diverse uses (i.e., birds themselves and birds as a link in the food chain). Concerning direct valuation in V-V approach, some limitations are related to the difficulties to calculate environmental non-use and option values [14].

Another limitation emerges from the functioning of evaluation processes and developers’ recommendations: make rigorous evaluations to improve knowledge and to reach reasonable results in economic and ecological perspectives. However, in this case, differences may appear when concerted agents emphasize the need for further and fairer assessments of destroyed biodiversity. These differences are mostly created by ratios chosen by experts during biodiversity inventories. Bruggeman and Jones [54] underlined that compensatory measures require a means to determine the appropriate level of off-site compensation and specify mitigation ratios. Indeed, assessing these ratios intend to capture dissimilarities in ecological functions between natural and created ecosystems [55]. In practice, these ratios are calculated by professionals, which means that the relation between these ratios and ecosystem functioning tends to be implicit and not clearly articulated [54]. Thus, ratios are leading experts to balance the importance of environmental damage with non-uniformed values on the environmental resilience capacity, time-life of species. In order to reduce the amount of offset, some developers take advantage of their dominant status in the negotiation of impact study realization. By this way, ratio decisions could largely be influenced by the economic interests of international firms who would like to pursue their development according to international recommendations.

9Biologists and ecologists determine these ratios. They include the scarcity, the size of the damage, the capacity of reconstruction, etc..
4. Conclusions

Biodiversity and ecosystem services largely contribute to human welfare, and this seems to be an evidence that human attribute values to these elements. One of the goals of biodiversity banking is to stop the state of the environment from deteriorating and to help building a future where human being will live in harmony with his surroundings. Valuation methodologies employed to implement biodiversity banks face lots of critics but should be first considered as dialog triggers. Indeed, even if these methods are often criticized, they have the privilege to remembering the importance of conserving biodiversity and ecosystem services. However, it should be dangerous systematically considering results as exact information because lots of criteria are chosen by specialists themselves to analyse their own concerns and questions or to justify their thesis. In this paper, most of the difficulties noted were related to difficulties in realizing precise and dynamic inventories and to the capacity to estimate the biodiversity evolutions and adapt compensatory measures to them.

Several methodologies are analysed for ecological and economic assessments. Service-to-service and resource-to-resource approaches are non-monetary valuations made by wildlife specialists and frequently used by non-commercial biodiversity banks. To contrast with these approaches, monetary tools such as value-to-value, value-to-cost and benefits transfers are respectively used by private, hybrid or public commercial biodiversity banks or public non-commercial banks (large scale) within decision-making process in addition of ecological assessments. Benefits transfers or values transfers are often used on “similar” biodiversity to reduce evaluation time.

The principal result of this paper deals with the distinction between these major methodologies evaluating ecological impacts of project development. In practice, biodiversity banks use ecological assessments to establish biodiversity units in non-commercial case and biodiversity banks use economic assessments to establish credits in commercial context. Economic methods were progressively implemented to complete ecological assessment. Even if scientific knowledge on biodiversity is in constant improvement, ecological assessments are still under discussions because of the lack of information and forecasts. Economic assessments should be considered separately to distinguish all their characteristics. V-V and V-C methodologies are both used to attribute economic value to biodiversity but the way to attribute the price is different (nature commodification vs. compensatory measure costs addition). Nowadays, the V-C methodology keeps all the attention to be able to make robust assessment of ecological impacts, e.g., to estimate the environmental damage and design adapted compensatory measures.

Consequently, this paper represents a significant contribution to biodiversity offset and especially to biodiversity banking analysis distinguishing banking schemes and their associated valuation methodologies. It illustrates how valuation works in biodiversity banking and explains methodologies applied in each banking scheme. Biodiversity banking is still in design and evolution in industrialized countries. This evolution leads to many misunderstandings and confusions that this paper tries to clarify within a critical analysis of methodologies evaluating biodiversity in offset banking.

To conclude, this paper notes that ecological assessments are still essential in biodiversity banking process to assess environmental damage and give advice to implement compensatory measures. This paper recommends more collaboration between actors from different disciplines and the reinforcement of interdisciplinary within academic research. For non-commercial banks, ecological assessments create biodiversity units in order to offset environmental damage caused by public entities or single-user.
Public non-commercial should pursue their effort in creating methodologies evaluating biodiversity (i.e., NOAA, Environmental Agencies, etc.). As concerns economic assessments, they should be based on ecological data to implement in-kind offsets. V-C methodology seems to be efficient to cover the costs of offsetting while offering a visible solution to developers. However, all actors should use with caution all these methodologies considering their opportunities and risks to avoid or reduce collateral damage.

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